Utah Lake Littoral Sediment Study: An Assessment of Carbon, Nitrogen, and Phosphorus Dynamics in the Utah Lake Littoral Zone

Literature Review

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Contents

1. Introduction

The Utah Division of Water Quality (DWQ) is in Phase 2 of the Utah Lake Water Quality Study (ULWQS) to evaluate the effect of excess nutrients on the lake's recreational, aquatic life, and agricultural designated uses and to develop site-specific nitrogen and phosphorus water quality criteria to protect these uses. The ULWQS is guided by the Stakeholder Process developed during Phase 1, which established a 16-member interest-based Steering Committee and a 10-member disciplinary-based Science Panel. The Steering Committee charged the Science Panel with developing and answering key questions to characterize historic, current, and future nutrient conditions in Utah Lake. Responses to the key questions will be used by the Steering Committee to establish management goals for the lake and by the Science Panel to guide development of nutrient criteria to support those goals.

There is a high degree of annual variability in water levels in Utah Lake (Figure 1; CUWCD and Thurin 2007). The fluctuations result from a combination of natural and anthropogenic factors, including variable precipitation patterns, evaporation, upstream water use, and managed outflows. On average, lake levels vary by ~3-4 feet per year (measured as the difference between the minimum and maximum water level). Due to the shallow nature of the lake, which averages 9 feet deep when full, the fluctuating water levels cause major changes in water-edge location and lake characteristics.

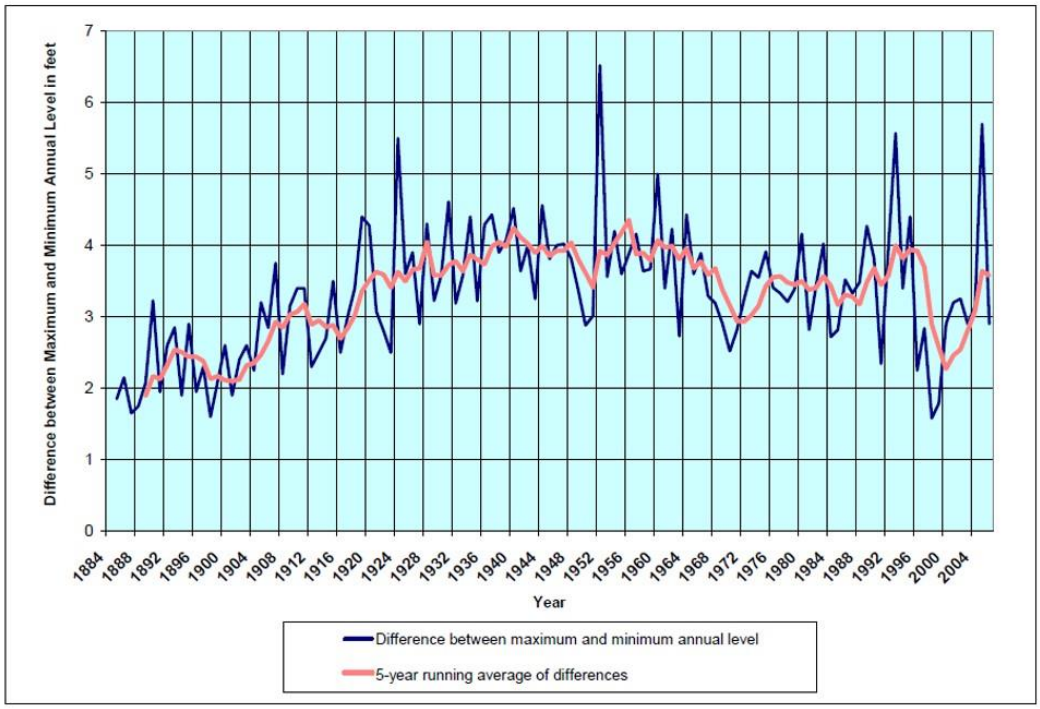

Figure 1. Annual and five-year average within-year variation in Utah Lake level from 1884 to 2006 showing generally increasing variation (doubling) over the historical period from 1884 to the 1930s to 1940 (from CUWCD and Thurin 2007; Figure 11).

With the large variations in lake levels and shallow depth, large expanses of Utah Lake's littoral sediments are subject to wetting and drying cycles of varying durations and frequencies.

Little is currently known about the effects of water level fluctuations/wet and dry phases on C, N, and P loading from littoral sediments in Utah Lake. As the Science Panel works to respond to charge questions and nutrient criteria are being developed, it is important that this knowledge gap be addressed. More specifically, the Science Panel needs to better understand whether littoral sediments act as nutrient sinks (e.g., through denitrification, respiration and sedimentation) or sources (e.g., decomposition/mineralization and release upon rewetting), to what magnitude, and whether they reduce or enhance nutrient loads and impact the overall nutrient budget of the lake. The Science Panel also needs quantitative relationships between the duration and frequency on wetting and drying on nutrient loading in order to evaluate relationships between external and internal loads to Utah Lake.

2. Previous Utah Lake studies and datasets

Water level data have been collected in Utah Lake since the late 1800's (Figure 1; CUWCD and Thurin 2007). A probability distribution of fluctuations in lake area using data from 2004-2018 estimated that the 5 to 95th percentile in lake area varied by 30 mi^2 from an average of 130 mi^2 (J. Martin, pers. comm). Based on other estimates, approximately 10-15% of the area is littoral. The extent of the areas of wetting and drying can also be illustrated by the Utah Lake bathymetry (Figure 2) comparing areas that were always wet to those that were periodically dry. In either case, the amount of lake area potentially affected by wetting and drying is substantial. The duration of dry and wet phases can also be inferred from lake level data and can range from months to years.

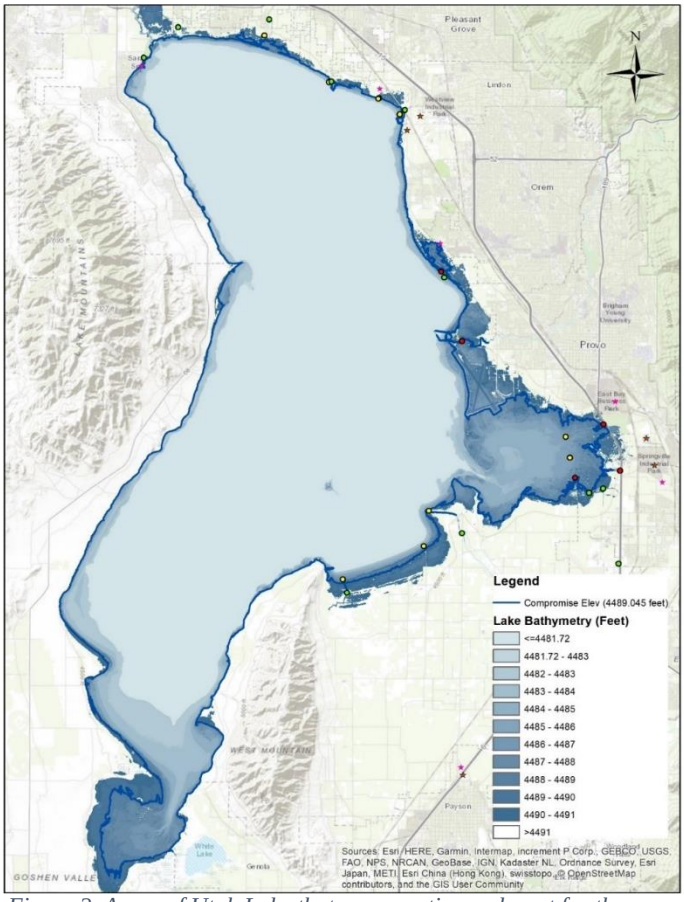

Figure 2. Areas of Utah Lake that were continuously wet for the period of 2010 – 2020 (light blue) versus those that periodically became dry (Dark blue bands).

Other existing, complementary studies include a project recently completed by Goel and Carling on

sediment–water-nutrient interactions in Utah Lake[1] (Goel et al., 2020). Results include calculations of sediment fluxes over a range of water column P concentrations and an exploration of the potential effects of changing pH, alkalinity, and redox. Equilibrium P concentration, the water column concentration at which the sediment switches from a sink to a source of P, were estimated from this study as well. The study also estimated sediment oxygen demand and information on the role of sediment resuspension on nutrient releases or removal, primarily via calcite scavenging. The experiments were performed on wet cores collected from two sites —one site in Provo Bay and one site from the main body of the lake at an established DWQ monitoring site. These data could be contrasted with the results from this work, but also provide information on wet core nutrient content.

2.1. Utah Lake drying and re-inundation events

We found no research detailing the release rates of N or P from Utah Lake littoral sediments. However, Utah Lake experiences lake water fluctuations (LWF) that cause sediments to dry and become re-inundated. Generally, dry littoral sediments may experience one re-inundation event as the lake level rises in the spring and declines in the fall or sediments that straddle the margin of the lake may experience multiple intra-annual fluctuations and drying-re-inundation event. Further, Utah Lake levels may also remain low over multiple years, leading to the re-inundation event of sediments that were previously saturated or periodically exposed to drying. The re-inundation of dry sediments or relatively dry sediments may release N and P and result in substantial nutrient loading. Aside from that, little is known about littoral sediments in Utah Lake, thus the need for this work.

3. Effects and mechanisms of sediment drying and rewetting on C, N, and P flux

It is well established that drying and rewetting events alter microbial activity and biogeochemical processes in sediments (Amalfitano et al., 2008; Baldwin and Mitchell, 2000; Larned et al., 2007; Marxsen et al., 2010; Pohlon et al., 2013; Zoppini and Marxsen, 2010). As littoral sediments go through periods of desiccation and inundation, it changes sediment properties and alters the duration of oxic and anoxic conditions, which in turn affects sediment oxygen demand, C, N and P release as well as microbial activity and composition (Weise et al., 2016). A number of studies have found that sediment drying promotes the release of potentially significant amounts of bio-available N and P on re-wetting (the so-called “Birch effect”; Baldwin and Mitchell, 2000; Birch, 1960; McComb and Qiu, 1998; Scholz et al., 2002). This occurs as a result of numerous interacting processes, including enhanced aerobic microbial mineralization of OM and the reduction of nitrate, leading to an accumulation of ammonium N in the sediment; a decreased capacity of the sediments to adsorb nutrients such as P (Baldwin, 1996); and the release of cell-bound nitrogen (ammonium) and filterable reactive phosphorus from sediment bacteria as they are killed during drying (Qiu and McComb, 1995). Although both N and P may be released by these processes, they may respond differently, since the re-wetted sediments may

[1]this study was performed on fully-wetted sediments, not sediments subjected to drying

have a reduced capacity to release P under anoxic conditions (which suggests that more N than P could be released into the water column on lake filling) (Mitchell and Baldwin, 1998). The degree and duration of drying before rewetting has been shown to affect nutrient release. Schönbrunner et al. (2012) performed an internal phosphorus loading study in which floodplain sediments were exposed to different dry/wet treatments. They found that total phosphorus (TP) release from sediments into the water column increased with increasing duration of dry periods prior to rewetting and that repeated drying and wetting resulted in elevated phosphorus release. This effect was more pronounced when drying periods led to an 80% reduction in water content.

Sediment characteristics also affect nutrient releases. Shaughnessy et al. (2019) found that spatial distributions of lakebed nutrients in an agricultural reservoir in Illinois were predominantly controlled by sediment depositional patterns. The largest proportion of clay-sized particles and highest concentrations of OM were deposited near the dam wall and the highest proportion of (heavier) sand-sized particles were deposited near the river mouth. They found a significant and positive correlation between TP, TN, and TC with OM. Shaughnessy et al. (2019) also found that seasonal factors were important to consider. Nitrogen species varied seasonally at the sediment-water interface and were significantly higher during warmer weather/the growing season. The warmer conditions may enhance the release of nutrients from the sediments to the water column due to higher decomposition rates, higher pH due to photosynthetic activities, and low DO near the sediment-water interface that can change redox conditions so that reduced iron (Fe) might liberate P.

Multiple factors determine the potential for nutrient loading across the sediment-water interface such as antecedent inorganic N and orthophosphate sediment concentrations, C:N:P ratio of organic matter comprising the sediments (Strauss and Lamberti, 2000), anthropogenic loading to sediments (Steinman et al 2014) and the microbial community (Conrad et al., 2014). However, the magnitude, duration, and frequency of LWF exerts a large influence over the biophysical parameters that regulate nutrient loading from the sediments to the water column.

3.1. C Flux

Local water use may influence GHG emissions to the atmosphere by exposing desiccated lake beds to air, as well as through changing lake water chemistry through lowering water levels. The CO_2 emissions from lake or stream beds exposed to the air by desiccation can be extremely high, making them an overlooked “blind spot” in global carbon cycling (Keller et al., 2020). Less is known about the response of landscape methane (CH_4) following desiccation (Keller et al., 2020), even though the global warming potentials of CH_4 is 34% greater than that of CO_2. Lake, stream, and reservoir sediments experiencing frequent re-wetting events and drawdown periods may indeed be major sources of CH_4 atmosphere (Jin et al., 2016; Kosten et al., 2018).

Declining water levels can influence water chemistry and biota in lakes (Callieri et al., 2014), and thus potentially alter aquatic GHG emissions as well. Quantifying the GHG emissions associated with incremental desiccation and local water management behaviors is thus not only important for understanding potential critical feedbacks with desiccation and drought within the

environmental system, but a better constraining of these GHG emissions may interact with societal perceptions of the risks associated with desiccation, and these perceptions may likewise change with local economic and societal conditions (Givens and Jorgenson, 2011).

3.2. N Flux

The nitrogen cycle is biologically mediated and composed of transformation pathways that provide sources and sinks for bioavailable N, and the regulation of these processes determines the net bioavailable N in the system (Berman and Bronk, 2003; Daly et al., 2021). Hydrologic fluctuations present an important control on sediment microorganisms and subsequent nitrogen (N) transformation rates (Gutknecht et al., 2006). Sediment desiccation draws down moisture content, thereby increasing oxygen concentrations and gas diffusivity through pore space while decreasing mobility of dissolved solids and osmotic potential of microorganisms. Processes within the N cycle respond to thermodynamic conditions of the sediment-water interface [i.e. supply and availability of electron donors (DOC, CH_4, NH_4) and acceptors (O_2, NO_3, SO_4)] which modulate the directionality and magnitude of N species production and consumption (Hedin et al., 1998).

3.2.1. Effects of sediment drying and rewetting on *sources* of bioavailable N

Processes within the N cycles that result in net production of N include N-fixation, ammonification/N-mineralization (organic N to inorganic N as NH_4), and microbial biomass turnover (N recycling). Nitrification and dissimilatory reduction of nitrate to ammonium are internal cycling processes that convert bioavailable N from one form to another (NH_4 to NO_3 and NO_3 to NH_4, respectively). All processes are sensitive to redox state and osmotic potential and have been observed to be influenced by hydrologic fluctuations in sediments in the following ways:

N-fixation

The impact of intermittent drying and wetting periods on benthic N-fixation has not yet been explicitly addressed in the literature.

Nitrification

Net nitrification has generally shown no significant change following dry-to-wet transitions, however, desiccation of saturated sediments has been shown to stimulate nitrification, resulting in decreased NH_4 but an increased buildup of NO_3 in dry sediments that can lead to a pulse of bioavailable N to the water column upon rewetting (Birch, 1960; Fromin et al., 2010; Gómez et al., 2012; Peralta et al., 2013). Nitrification has been observed to be particularly significant during later stages of drying periods (30+ days), and it is suggested that periodic drying and rewetting of littoral sediments reduces the potential for NH_4 release (Liu et al., 2019).

N-mineralization

Net mineralization increases during drying phases, particularly in the initial 10-25 days of the drying period (Gómez et al., 2012; Liu et al., 2019). Intermittent drying enhances the amount of mineralized N produced, and such wet-to-dry and dry-to-wet conditions can supply more N than under constant moist or saturated conditions (Birch, 1964).

Dissimilatory nitrate reduction to ammonium

DNRA is a significant, but understudied, process that produces NH_4 under anaerobic or anoxic conditions. Similar to denitrification, DNRA decreases in drying sediments, with a reduction in rates up to 90% (Rahman et al., 2019). DNRA has been observed to follow peak denitrification activity in the progression of sediment rewetting with decreasing redox conditions (Figure 3; Arce et al., 2015). DNRA may be a significant contributing factor to water column NO_3 removal during saturated conditions (Arce et al., 2015). Studies that have only considered changing water column NO_3 concentrations in response to sediment hydrologic fluctuations may have incorrectly attributed this NO_3 depletion to denitrification, and thus, overestimated the potential for N removal from the system. Temperature has been observed to be a key factor in determining the partition between denitrification and DNRA, indicating that temperatures may drive whether bioavailable N is removed from the system or recycled internally (Rahman et al., 2019).

Microbial biomass turnover

Sediment rewetting is known to enhance sediment N content, in part due to the abrupt osmotic shift and the resulting lysis of microbial cells upon rewetting (Birch, 1960). The contribution of microbial biomass turnover to N release during sediment drying and rewetting has not been explicitly measured, but it is assumed that NH_4 accumulation and release may be a result of microbial biomass killed during sediment drying (Qiu and McComb, 1995).

3.2.2. Effects of sediment drying and rewetting on *sinks* for bioavailable N

N cycle processes that result in net consumption or removal of N include denitrification (NO_3 to N_2), immobilization or assimilation (NH_4 or NO_3 to organic-N), volatilization (NH_4 to NH_3 gas), and leaching (hydrologic transport). Studies have largely only investigated the influence of hydrologic fluctuations on sediment denitrification.

Denitrification

Denitrification reduces nitrate to atmospheric N in the absence of oxygen and is the only known permanent removal mechanism for N in aquatic environments. Sediment denitrification is highly sensitive to the redox state, as the transformation only occurs when oxygen is not present to act as the electron acceptor in microbial metabolic respiration, and microbes instead utilize nitrate in the place of oxygen. Several studies have evaluated the response of sediment denitrification to hydrologic fluctuations across aquatic habitats, including streams, wetlands, ponds, and lakes. In general, it has been found that rewetting desiccated sediments stimulates increased rates of denitrification due to rapid depletion of oxygen (Arce et al., 2015; Fromin et al., 2010). Arce et al. (2015) observed maximum rates of denitrification occurring during the 24 hours following the

start of the rewetting period and a total decrease in water column nitrate by 94% in the week following rewetting.

Denitrification tends to decrease with decreasing sediment water content due to increased oxygen diffusivity, and thus denitrification generally decreases in drying sediments (Gómez et al., 2012; Koschorreck, 2005; Rahman et al., 2019). However, some studies have observed a decrease in net sediment inorganic N during sediment drying periods which may be a result of tightly coupled nitrification-denitrification (Koschorreck, 2005; Pinto et al., 2021)

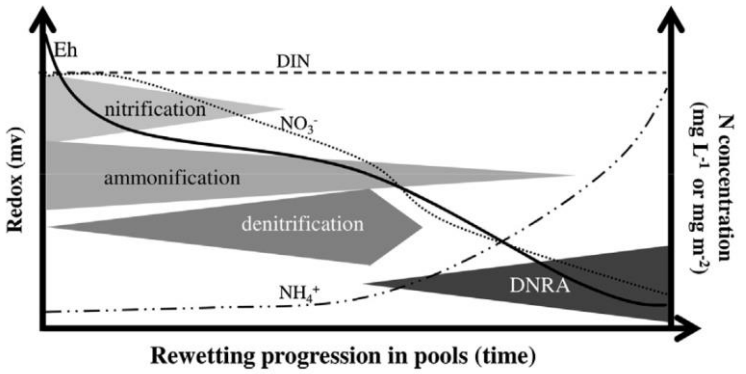

Figure 3. Hypothesized changes in the N cycle as sediments undergo rewetting following a period of desiccation (Arce et al., 2015)

3.2.3. Littoral sediments and N cycling focusing on N available to harmful algal bloom**s**

Bacterial activity may be extremely low in dry sediments owing to a combination of desiccation stress and reduced diffusion of substrates (Schimel et al., 2007). Under these relatively inactive conditions, precipitation events or LWF may serve as an environmental cue that terminates microbial dormancy under dry conditions and stimulates bacterial activity (Aanderud et al., 2015). In lakes during sediment re-inundation, bacteria from littoral sediments may release immense amounts of N and P creating a large source of nutrient loading (Søndergaard et al., 2013; Wu et al., 2017). For example, even in oligotrophic lake sediments exposed to LWF, the release rate of TN increased 30-fold (-1.4-1.1 mg m^{-2} d^{-1} permanently flooded, 33.4 to 52.7 mg m^{-2} d^{-1} exposed to LWF) and TP increased 20-fold (0.007 to 0.03 mg m^{-2} d^{-1} flooded, 0.6 to 1.4 mg m^{-2} d^{-1} exposed to LWF) over 75 days (Paranaíba et al., 2020). The resulting N and P entering the water column across the sediment-water interface may be directly used by harmful cyanobacteria and algae (Bormans et al., 2016; Glibert et al., 2016; Jeppesen et al., 2005).

3.3. P Flux

In nearly all studies investigating wetting/drying dynamics of soils, reservoir sediments, river sediments, and lake sediments, P is released to the overlying water during the rewetting phase. The amount of P flux depends on the drying history of the sediments, with littoral often having a higher P flux relative to deeper (always inundated) lake sediments (Andersen and Ring, 1999; de

Vicente et al., 2010; Steinman et al., 2012). The amount of drying is important, with fully desiccated sediment releasing more P than partially dry sediments (Baldwin, 1996; Baldwin and Mitchell, 2000). Periodic wetting and drying may cause an increase in P flux as opposed to only one wetting/drying cycle (Liu et al., 2019; Schönbrunner et al., 2012). Repeated drying and wetting may result in increased P flux from sediments (Schönbrunner et al., 2012). Rewetted sediments may be a short-term P source to the overlying water before the P flux gradually decreases over time since water was added (Tang et al., 2014; Thanh Nguyen and Marschner, 2005). An increased P release is related to a decrease in P sorption capacity of soils and sediments (Attygalla et al., 2016; de Vicente et al., 2010; Gilbert et al., 2014; Qiu and McComb, 2002). The increase in P concentrations during rewetting is often greater in porewater relative to overlying water (Ding et al., 2019; Kinsman-Costello et al., 2016; Tian et al., 2017; Young and Ross, 2001). The amount of P release from sediments upon rewetting may be related to the fraction of loosely bound P from sequential extraction tests (Zhang et al., 2012).

4. Mechanisms of C, N, and P flux during sediment drying and rewetting phases

The dynamics of hydrologic shifts, including the frequency, intensity, and duration of dry-wet cycles affects microbial N responses (Foulquier et al., 2013; Langhans and Tockner, 2006; Larned et al., 2007).

4.1. Magnitude: lake re-inundation versus rainfall on sediments

The impact of a rewetting event on N is determined by the degree of saturation caused by the lake re-inundation or rainfall event. The level of dissolved oxygen (DO) in sediments may regulate the form of inorganic N released during rewetting. For example, in an agriculturally impacted stream, rewetting of sediments increased NH_4 concentration over a seven-day rewetting, which suggested that NO_3 was transformed into NH_4 through DNRA supported by low Eh levels (40 mV; Arce et al., 2015). Alternatively, rainfall on dry sediments or more air-exposed sediments stimulated ammonium oxidation to NO_3 six to nine weeks after a rainfall event (22 mm; Arce et al., 2018). In general, oxic conditions reduce anaerobic processes such as denitrification (Austin and Strauss, 2011; Gómez et al., 2012) and favor aerobic pathways such as nitrification. In Utah Lake, sediments rather than the water column are typically a sink for DO (Hogsett et al., 2019). The water column is commonly supersaturated with DO regardless of location most likely due to the primary production by phytoplankton in the spring, summer, and fall (Lawson et al 2021) and wind and wave action continual mixing of the water column.

4.2. Duration: N release in wet and dry lake sediments

LWF may create drying rewetting regimes that vary in duration. For example, the net release of N and P into water from air-dried sediment increases after re-inundation (Reddy et al., 1999; Steinman et al., 2012), but the duration of re-inundation determined the response. Steinman et al (2012) revealed that the fluxes of both soluble reactive phosphorus (SRP) and NH_4 from dried wetland sediment were influenced when long-term drying was followed by short-term re-inundation. Also, following re-inundation in sediments from the shallow eutrophic Lake Chaohu,

China, NH_4 fluxes increased following 5-25 days of drying, which was followed by an obvious decrease after 30 days of drying (Liu et al., 2019). The decreased of NH_4 fluxes remained low during the remaining 90 days re-inundation period as well. In contrast to re-inundation sediment drying also stimulates the release of N. For example, during the first 10 days of sediments drying in a semi-arid intermittent stream, desiccation stimulated net N-mineralization and net nitrification from sediments (Gómez et al., 2012). However, as drying continued much of the inorganic N potentially vanishes. In sediments from an Amazonian lake floodplain, N loss was curbed by the inhibition of nitrification, suggesting that coupled nitrification-denitrification was responsible for N loss and not microbial assimilation or leaching (Koschorreck, 2005).

4.3. The frequency of the re-inundation may determine the release of N and P

In a large shallow eutrophic lake with high WLF, drying and re-inundation events may only occur once annually or occur multiple times in a season predominantly due to the location of the sediment. If the sediment is near the edge of the lake, the intra-annual variation in lake level caused by LWF may create multiple drying and re-inundation cycles. For example, in river sediments on the bench that flood every year compared to sediments permanently saturated in the lakebed, concentrations of NH_4 were lower but NO_3 and SRP were higher (Woodward et al., 2015). In Utah Lake sediments that were consistently inundated by water, NH_4 fluxes were negative demonstrating a loss of N (Goel et al., 2020). Sediments exposed to oxic conditions generally lost more NH_4 (-33.4 to -3.6 mg m-2 d-1) than sediments that were slightly anoxic (-9.12 to 0.11 mg m-2 d-1). Generally, NH_4 but not NO_3 was detectable in sediments. The loss of NH_4 was potentially due to a source-sink relationship between microorganisms sequestering N in the water column, ammonia volatilization due to relatively high pH levels, and/or ammonium oxidation and coupled nitrification-denitrification (Goel et al., 2020; Palmer et al., 2009). Further, in Provo Bay, Utah Lake, the flux of N from sediments was immense, -1.44 g N m-2 d-1, demonstrating this location's potentially high sediment N mineralization rates resulting from organic matter decomposition or high dissimilatory nitrate reduction to ammonium activities under higher-oxygen depletion conditions (Hogsett et al., 2019).

The mechanisms of P release during drying and wetting sediments include: 1) reductive dissolution of iron during anoxic conditions during wetting (Andersen and Ring, 1999; Schönbrunner et al., 2012; Young and Ross, 2001); a change in P speciation and increase in reactive P as organic P is mineralized during drying (Attygalla et al., 2016; Kerr et al., 2010; Liu et al., 2019); a decrease in P adsorption capacity of sediment during drying as iron speciation shifts to more recalcitrant forms or organic matter is lost drying; death of bacteria during desiccation of sediments causing P release (Baldwin and Mitchell, 2000; Turner and Haygarth, 2001); decay of plants (Keitel et al., 2016; Klamt et al., 2016); and change in sediment mineralogy (Baldwin and Mitchell, 2000) or particle size distribution (Tang et al., 2014) during drying. The mechanisms for P release during wetting and drying cycles can be summarized as chemical reactions dominated by sorption-desorption processes controlled by pH and redox conditions; physical transformations such as changes in particle size distribution; and biological processes dominated by microbial activities (de Vicente et al., 2010). Hydrological processes also play a role as P exchanges from pore water to the water column with frequent wetting and

drying cycles or water level fluctuations (Ding et al., 2019; Kinsman-Costello et al., 2016; Steinman et al., 2014, 2012; Tang et al., 2016).

5. Synthesis of peer-reviewed field and laboratory protocols

The methods for quantifying or explaining P and N release from sediments to the water column during drying and rewetting include a number of laboratory experiments including: 1) wetting/drying intact sediment cores along transects from dry to wet sediments (Andersen and Ring, 1999; Ding et al., 2019; Klamt et al., 2016; Liu et al., 2019; Paranaíba et al., 2020; Qiu and McComb, 1994; Schönbrunner et al., 2012; Steinman et al., 2014, 2012; Tang et al., 2016); conducting sequential extractions or batch sorption tests on sediment samples with different amounts of drying to test for P speciation and adsorption capacity (Attygalla et al., 2016; Baldwin, 1996, p. 199; de Vicente et al., 2010; Gilbert et al., 2014; Kerr et al., 2010; Qiu and McComb, 2002; Turner and Haygarth, 2001; Zhang et al., 2012); creating sediment/water slurries in centrifuge tubes and conducting wetting/drying experiments (Baldwin and Mitchell, 2000; Mitchell and Baldwin, 1998); and flooding sediment samples in columns or mesocosms (Dieter et al., 2015; Gerhardt et al., 2010; Keitel et al., 2016; Kinsman-Costello et al., 2016; Tang et al., 2016; Thanh Nguyen and Marschner, 2005; Tian et al., 2017; Watts, 2000; Wilson and Baldwin, 2008; Young and Ross, 2001). Several studies measured P concentrations in porewater in addition to overlying water during the wetting phase (Ding et al., 2019; Tian et al., 2017; Young and Ross, 2001). One study investigated P release through field observations by measuring water column P concentrations during flooding of historically drained wetlands (Kinsman-Costello et al., 2016).

6. References

Aanderud, Z.T., Jones, S.E., Fierer, N., Lennon, J.T., 2015. Resuscitation of the rare biosphere contributes to pulses of ecosystem activity. Front. Microbiol. 6. https://doi.org/10.3389/fmicb.2015.00024

Amalfitano, S., Fazi, S., Zoppini, A., Caracciolo, A.B., Grenni, P., Puddu, A., 2008. Responses of benthic bacteria to experimental drying in sediments from Mediterranean temporary rivers. Microb. Ecol. 55, 270–279.

Andersen, F.Ø., Ring, P., 1999. Comparison of phosphorus release from littoral and profundal sediments in a shallow, eutrophic lake, in: Walz, N., Nixdorf, B. (Eds.), Shallow Lakes ’98. Springer Netherlands, Dordrecht, pp. 175–183. https://doi.org/10.1007/978-94-017-2986-4_19

Arce, M.I., Sánchez-Montoya, M. del M., Gómez, R., 2015. Nitrogen processing following experimental sediment rewetting in isolated pools in an agricultural stream of a semiarid region. Ecol. Eng. 77, 233–241. https://doi.org/10.1016/j.ecoleng.2015.01.035

Arce, M.I., von Schiller, D., Bengtsson, M.M., Hinze, C., Jung, H., Alves, R.J.E., Urich, T., Singer, G., 2018. Drying and Rainfall Shape the Structure and Functioning of Nitrifying Microbial Communities in Riverbed Sediments. Front. Microbiol. 9, 2794. https://doi.org/10.3389/fmicb.2018.02794

Attygalla, N.W., Baldwin, D.S., Silvester, E., Kappen, P., Whitworth, K.L., 2016. The severity of sediment desiccation affects the adsorption characteristics and speciation of phosphorus. Environ. Sci. Process. Impacts 18, 64–71. https://doi.org/10.1039/C5EM00523J

Austin, B.J., Strauss, E.A., 2011. Nitrification and denitrification response to varying periods of desiccation and inundation in a western Kansas stream. Hydrobiologia 658, 183–195.

Baldwin, D.S., 1996. Effects of exposure to air and subsequent drying on the phosphate sorption characteristics of sediments from a eutrophic reservoir. Limnol. Oceanogr. 41, 1725–1732. https://doi.org/10.4319/lo.1996.41.8.1725

Baldwin, D.S., Mitchell, A., 2000. The effects of drying and re-flooding on the sediment and soil nutrient dynamics of lowland river–floodplain systems: a synthesis. Regul. Rivers Res. Manag. Int. J. Devoted River Res. Manag. 16, 457–467.

Berman, T., Bronk, D., 2003. Dissolved organic nitrogen: a dynamic participant in aquatic ecosystems. Aquat. Microb. Ecol. 31, 279–305. https://doi.org/10.3354/ame031279

Birch, H., 1960. Nitrification in soils after different periods of dryness. Plant Soil 12, 81–96.

Birch, H.F., 1964. Mineralisation of plant nitrogen following alternate wet and dry conditions. Plant Soil 20, 43–49. https://doi.org/10.1007/BF01378096

Bormans, M., Maršálek, B., Jančula, D., 2016. Controlling internal phosphorus loading in lakes by physical methods to reduce cyanobacterial blooms: a review. Aquat. Ecol. 50, 407–422.

Callieri, C., Bertoni, R., Contesini, M., Bertoni, F., 2014. Lake level fluctuations boost toxic cyanobacterial “oligotrophic blooms.” PLoS ONE 9, 1–8. https://doi.org/10.1371/journal.pone.0109526

Conrad, R., Ji, Y., Noll, M., Klose, M., Claus, P., Enrich-Prast, A., 2014. Response of the methanogenic microbial communities in Amazonian oxbow lake sediments to desiccation stress: Methanogenic microbial communities and desiccation stress. Environ. Microbiol. 16, 1682–1694. https://doi.org/10.1111/1462-2920.12267

Daly, A.B., Jilling, A., Bowles, T.M., Buchkowski, R.W., Frey, S.D., Kallenbach, C.M., Keiluweit, M., Mooshammer, M., Schimel, J.P., Grandy, A.S., 2021. A holistic framework integrating plant-microbe-mineral regulation of soil bioavailable nitrogen. Biogeochemistry 154, 211–229. https://doi.org/10.1007/s10533-021-00793-9

de Vicente, I., Andersen, F.Ø., Hansen, H.C.B., Cruz-Pizarro, L., Jensen, H.S., 2010. Water level fluctuations may decrease phosphate adsorption capacity of the sediment in oligotrophic high mountain lakes. Hydrobiologia 651, 253–264. https://doi.org/10.1007/s10750-010-0304-x

Dieter, D., Herzog, C., Hupfer, M., 2015. Effects of drying on phosphorus uptake in re-flooded lake sediments. Environ. Sci. Pollut. Res. 22, 17065–17081. https://doi.org/10.1007/s11356-015-4904-x

Ding, J., Hua, Z., Chu, K., 2019. The effect of hydrodynamic forces of drying/wetting cycles on the release of soluble reactive phosphorus from sediment. Environ. Pollut. 252, 992–1001. https://doi.org/10.1016/j.envpol.2019.06.016

Foulquier, A., Volat, B., Neyra, M., Bornette, G., Montuelle, B., 2013. Long-term impact of hydrological regime on structure and functions of microbial communities in riverine wetland sediments. FEMS Microbiol. Ecol. 85, 211–226.

Fromin, N., Pinay, G., Montuelle, B., Landais, D., Ourcival, J.M., Joffre, R., Lensi, R., 2010. Impact of seasonal sediment desiccation and rewetting on microbial processes involved in greenhouse gas emissions. Ecohydrology 3, 339–348. https://doi.org/10.1002/eco.115

Gerhardt, S., Boos, K., Schink, B., 2010. Uptake and release of phosphate by littoral sediment of a freshwater lake under the influence of light or mechanical perturbation. J. Limnol. 69, 54. https://doi.org/10.4081/jlimnol.2010.54

Gilbert, J.D., Guerrero, F., de Vicente, I., 2014. Sediment desiccation as a driver of phosphate availability in the water column of Mediterranean wetlands. Sci. Total Environ. 466–467, 965–975. https://doi.org/10.1016/j.scitotenv.2013.07.123

Givens, J.E., Jorgenson, A.K., 2011. The effects of Affluence, Economic Development, and Environmental Degradation on Environmental Concern: A multilevel analysis. Organ. Environ. 24, 74–91. https://doi.org/10.1177/1086026611406030

Glibert, P.M., Wilkerson, F.P., Dugdale, R.C., Raven, J.A., Dupont, C.L., Leavitt, P.R., Parker, A.E., Burkholder, J.M., Kana, T.M., 2016. Pluses and minuses of ammonium and nitrate uptake and assimilation by phytoplankton and implications for productivity and community composition, with emphasis on nitrogen-enriched conditions: Pluses and minuses of NH4+ *and* NO3−. Limnol. Oceanogr. 61, 165–197. https://doi.org/10.1002/lno.10203

Goel, R., Carling, G., Li, H., Smithson, S., 2020. Utah Lake sediment–water nutrient interactions. Utah Department of Environmental Quality.

Gómez, R., Arce, M.I., Sánchez, J.J., del Mar Sánchez-Montoya, M., 2012. The effects of drying on sediment nitrogen content in a Mediterranean intermittent stream: a microcosms study. Hydrobiologia 679, 43–59. https://doi.org/10.1007/s10750-011-0854-6

Gutknecht, J.L.M., Goodman, R.M., Balser, T.C., 2006. Linking soil process and microbial ecology in freshwater wetland ecosystems. Plant Soil 289, 17–34. https://doi.org/10.1007/s11104-006-9105-4

Hedin, L.O., von Fischer, J.C., Ostrom, N.E., Kennedy, B.P., Brown, M.G., Robertson, G.P., 1998. THERMODYNAMIC CONSTRAINTS ON NITROGENTRANSFORMATIONS AND OTHER BIOGEOCHEMICALPROCESSES AT SOIL–STREAM INTERFACES. Ecology 79, 684–703. https://doi.org/10.1890/0012-9658(1998)079[0684:TCONAO]2.0.CO;2

Hogsett, M., Li, H., Goel, R., 2019. The Role of Internal Nutrient Cycling in a Freshwater Shallow Alkaline Lake. Environ. Eng. Sci. 36, 551–563. https://doi.org/10.1089/ees.2018.0422

Jeppesen, E., Søndergaard, M., Jensen, J.P., Havens, K.E., Anneville, O., Carvalho, L., Coveney, M.F., Deneke, R., Dokulil, M.T., Foy, B., 2005. Lake responses to reduced nutrient loading–an analysis of contemporary long-term data from 35 case studies. Freshw. Biol. 50, 1747–1771.

Jin, H., Yoon, T.K., Lee, S.H., Kang, H., Im, J., Park, J.H., 2016. Enhanced greenhouse gas emission from exposed sediments along a hydroelectric reservoir during an extreme drought event. Environ. Res. Lett. 11. https://doi.org/10.1088/1748-9326/11/12/124003

Keitel, J., Zak, D., Hupfer, M., 2016. Water level fluctuations in a tropical reservoir: the impact of sediment drying, aquatic macrophyte dieback, and oxygen availability on phosphorus mobilization. Environ. Sci. Pollut. Res. 23, 6883–6894. https://doi.org/10.1007/s11356-015-5915-3

Keller, P.S., Catalán, N., von Schiller, D., Grossart, H.P., Koschorreck, M., Obrador, B., Frassl, M.A., Karakaya, N., Barros, N., Howitt, J.A., Mendoza-Lera, C., Pastor, A., Flaim, G., Aben, R., Riis, T., Arce, M.I., Onandia, G., Paranaíba, J.R., Linkhorst, A., del Campo, R., Amado, A.M., Cauvy-Fraunié, S., Brothers, S., Condon, J., Mendonça, R.F., Reverey, F., Rõõm, E.I., Datry, T., Roland, F., Laas, A., Obertegger, U., Park, J.H., Wang, H., Kosten, S., Gómez, R., Feijoó, C., Elosegi, A., Sánchez-Montoya, M.M., Finlayson, C.M., Melita, M., Oliveira Junior, E.S., Muniz, C.C., Gómez-Gener, L., Leigh, C., Zhang, Q., Marcé, R., 2020. Global CO2 emissions from dry inland waters share common drivers across ecosystems. Nat. Commun. 11, 1–8. https://doi.org/10.1038/s41467-020-15929-y

Kerr, J.G., Burford, M., Olley, J., Udy, J., 2010. The effects of drying on phosphorus sorption and speciation in subtropical river sediments. Mar. Freshw. Res. 61, 928. https://doi.org/10.1071/MF09124

Kinsman-Costello, L.E., Hamilton, S.K., O'Brien, J.M., Lennon, J.T., 2016. Phosphorus release from the drying and reflooding of diverse shallow sediments. Biogeochemistry 130, 159–176. https://doi.org/10.1007/s10533-016-0250-4

Klamt, A.-M., Reitzel, K., Andersen, F.Ø., Jensen, H.S., 2016. Simulated drawdown and rewetting of littoral sediments: implications for Lobelia lake management. Hydrobiologia 778, 137–150. https://doi.org/10.1007/s10750-015-2619-0

Koschorreck, M., 2005. Nitrogen Turnover in Drying Sediments of an Amazon Floodplain Lake. Microb. Ecol. 49, 567–577. https://doi.org/10.1007/s00248-004-0087-6

Kosten, S., van den Berg, S., Mendonça, R., Paranaíba, J.R., Roland, F., Sobek, S., Van Den Hoek, J., Barros, N., 2018. Extreme drought boosts CO2 and CH4 emissions from reservoir drawdown areas. Inland Waters 8, 329–340. https://doi.org/10.1080/20442041.2018.1483126

Langhans, S.D., Tockner, K., 2006. The role of timing, duration, and frequency of inundation in controlling leaf litter decomposition in a river-floodplain ecosystem (Tagliamento, northeastern Italy). Oecologia 147, 501–509.

Larned, S.T., Datry, T., Robinson, C.T., 2007. Invertebrate and microbial responses to inundation in an ephemeral river reach in New Zealand: effects of preceding dry periods. Aquat. Sci. 69, 554–567.

Liu, C., Du, Y., Chen, K., Ma, S., Chen, B., Lan, Y., 2019. Contrasting exchanges of nitrogen and phosphorus across the sediment–water interface during the drying and re-inundation of littoral eutrophic sediment. Environ. Pollut. 255, 113356. https://doi.org/10.1016/j.envpol.2019.113356

Marxsen, J., Zoppini, A., Wilczek, S., 2010. Microbial communities in streambed sediments recovering from desiccation. FEMS Microbiol. Ecol. 71, 374–386.

McComb, A., Qiu, S., 1998. The effects of drying and reflooding on nutrient release from wetland sediments. Environment Australia.

Mitchell, A., Baldwin, D.S., 1998. Effects of desiccation/oxidation on the potential for bacterially mediated P release from sediments. Limnol. Oceanogr. 43, 481–487. https://doi.org/10.4319/lo.1998.43.3.0481

Palmer, H., Beutel, M., Gebremariam, S., 2009. High rates of ammonia removal in experimental oxygen-activated nitrification wetland mesocosms. J. Environ. Eng. 135, 972–979. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000053

Paranaíba, J.R., Quadra, G., Josué, I.I.P., Almeida, R.M., Mendonça, R., Cardoso, S.J., Silva, J., Kosten, S., Campos, J.M., Almeida, J., Araújo, R.L., Roland, F., Barros, N., 2020. Sediment drying-rewetting cycles enhance greenhouse gas emissions, nutrient and trace element release, and promote water cytogenotoxicity. PLOS ONE 15, e0231082. https://doi.org/10.1371/journal.pone.0231082

Peralta, A.L., Ludmer, S., Kent, A.D., 2013. Hydrologic history influences microbial community composition and nitrogen cycling under experimental drying/wetting treatments. Soil Biol. Biochem. 66, 29–37. https://doi.org/10.1016/j.soilbio.2013.06.019

Pinto, R., Weigelhofer, G., Brito, A.G., Hein, T., 2021. Effects of dry-wet cycles on nitrous oxide emissions in freshwater sediments: a synthesis. PeerJ 9, e10767. https://doi.org/10.7717/peerj.10767

Pohlon, E., Ochoa Fandino, A., Marxsen, J., 2013. Bacterial community composition and extracellular enzyme activity in temperate streambed sediment during drying and rewetting. PloS One 8, e83365.

Qiu, S., McComb, A., 2002. Interrelations between iron extractability and phosphate sorption in reflooded air-dried sediments. Hydrobiologia 472, 39–44.

Qiu, S., McComb, A., 1995. Planktonic and microbial contributions to phosphorus release from fresh and air-dried sediments. Mar. Freshw. Res. 46, 1039–1045.

Qiu, S., McComb, A., 1994. Effects of oxygen concentration on phosphorus release from reflooded air-dried wetland sediments. Mar. Freshw. Res. 45, 1319. https://doi.org/10.1071/MF9941319

Rahman, M., Grace, M.R., Roberts, K.L., Kessler, A.J., Cook, P.L.M., 2019. Effect of temperature and drying-rewetting of sediments on the partitioning between denitrification and DNRA in constructed urban stormwater wetlands. Ecol. Eng. 140, 105586. https://doi.org/10.1016/j.ecoleng.2019.105586

Reddy, K.R., Kadlec, R.H., Flaig, E., Gale, P.M., 1999. Phosphorus retention in streams and wetlands: A review. Crit. Rev. Environ. Sci. Technol. 29, 83–146. https://doi.org/10.1080/10643389991259182

Scholz, O., Gawne, B., Ebner, B., Ellis, I., 2002. The effects of drying and re-flooding on nutrient availability in ephemeral deflation basin lakes in western New South Wales, Australia. River Res. Appl. 18, 185–196.

Schönbrunner, I.M., Preiner, S., Hein, T., 2012. Impact of drying and re-flooding of sediment on phosphorus dynamics of river-floodplain systems. Sci. Total Environ. 432, 329–337. https://doi.org/10.1016/j.scitotenv.2012.06.025

Shaughnessy, A., Sloan, J., Corcoran, M., Hasenmueller, E., 2019. Sediments in agricultural reservoirs act as sinks and sources for nutrients over various timescales. Water Resour. Res. 55, 5985–6000.

Søndergaard, M., Bjerring, R., Jeppesen, E., 2013. Persistent internal phosphorus loading during summer in shallow eutrophic lakes. Hydrobiologia 710, 95–107.

Steinman, A.D., Ogdahl, M.E., Weinert, M., Thompson, K., Cooper, M.J., Uzarski, D.G., 2012. Water level fluctuation and sediment–water nutrient exchange in Great Lakes coastal wetlands. J. Gt. Lakes Res. 38, 766–775. https://doi.org/10.1016/j.jglr.2012.09.020

Steinman, A.D., Ogdahl, M.E., Weinert, M., Uzarski, D.G., 2014. Influence of water-level fluctuation duration and magnitude on sediment–water nutrient exchange in coastal wetlands. Aquat. Ecol. 48, 143–159. https://doi.org/10.1007/s10452-014-9472-5

Strauss, E.A., Lamberti, G.A., 2000. Regulation of nitrification in aquatic sediments by organic carbon. Limnol. Oceanogr. 45, 1854–1859. https://doi.org/10.4319/lo.2000.45.8.1854

Tang, X., Wu, M., Li, Q., Lin, L., Zhao, W., 2014. Impacts of water level regulation on sediment physic-chemical properties and phosphorus adsorption–desorption behaviors. Ecol. Eng. 70, 450–458. https://doi.org/10.1016/j.ecoleng.2014.06.022

Tang, Y., Van Kempen, M.M.L., Van der Heide, T., Manschot, J.J.A., Roelofs, J.G.M., Lamers, L.P.M., Smolders, A.J.P., 2016. A tool for easily predicting short-term phosphorus mobilization from flooded soils. Ecol. Eng. 94, 1–6. https://doi.org/10.1016/j.ecoleng.2016.05.046

Thanh Nguyen, B., Marschner, P., 2005. Effect of drying and rewetting on phosphorus transformations in red brown soils with different soil organic matter content. Soil Biol. Biochem. 37, 1573–1576. https://doi.org/10.1016/j.soilbio.2005.01.015

Tian, J., Dong, G., Karthikeyan, R., Li, L., Harmel, R., 2017. Phosphorus dynamics in long-term flooded, drained, and reflooded soils. Water 9, 531. https://doi.org/10.3390/w9070531

Turner, B.L., Haygarth, P.M., 2001. Phosphorus solubilization in rewetted soils. Nature 411, 258–258. https://doi.org/10.1038/35077146

Watts, C.J., 2000. Seasonal phosphorus release from exposed, re-inundated littoral sediments of two Australian reservoirs. Hydrobiologia 431, 27–39.

Weise, L., Ulrich, A., Moreano, M., Gessler, A., E. Kayler, Z., Steger, K., Zeller, B., Rudolph, K., Knezevic-Jaric, J., Premke, K., 2016. Water level changes affect carbon turnover and microbial community composition in lake sediments. FEMS Microbiol. Ecol. 92, fiw035.

Wilson, J.S., Baldwin, D.S., 2008. Exploring the 'Birch effect' in reservoir sediments: influence of inundation history on aerobic nutrient release. Chem. Ecol. 24, 379–386. https://doi.org/10.1080/02757540802497582

Woodward, K.B., Fellows, C.S., Mitrovic, S.M., Sheldon, F., 2015. Patterns and bioavailability of soil nutrients and carbon across a gradient of inundation frequencies in a lowland river channel, Murray–Darling Basin, Australia. Agric. Ecosyst. Environ. 205, 1–8. https://doi.org/10.1016/j.agee.2015.02.019

Wu, Z., Liu, Y., Liang, Z., Wu, S., Guo, H., 2017. Internal cycling, not external loading, decides the nutrient limitation in eutrophic lake: A dynamic model with temporal Bayesian hierarchical inference. Water Res. 116, 231–240. https://doi.org/10.1016/j.watres.2017.03.039

Young, E.O., Ross, D.S., 2001. Phosphate Release from Seasonally Flooded Soils: A Laboratory Microcosm Study. J. Environ. Qual. 30, 91–101. https://doi.org/10.2134/jeq2001.30191x

Zhang, B., Fang, F., Guo, J., Chen, Y., Li, Z., Guo, S., 2012. Phosphorus fractions and phosphate sorption-release characteristics relevant to the soil composition of water-level-fluctuating zone of Three Gorges Reservoir. Ecol. Eng. 40, 153–159. https://doi.org/10.1016/j.ecoleng.2011.12.024

Zoppini, A., Marxsen, J., 2010. Importance of extracellular enzymes for biogeochemical processes in temporary river sediments during fluctuating dry–wet conditions, in: Soil Enzymology. Springer, pp. 103–117.